**Title:** The ecological impact of dumping massive amounts of *Sargassum* in beach and forest ecosystems

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**Abstract**

Anthropogenic activities and urbanization have led to increased nutrient runoff into coastal waters, causing macroalgae blooms globally. Consequently, there has been a rise in the frequency and scale of pelagic *Sargassum* strandings along the coastlines in the Caribbean, Gulf of Mexico, and West Africa, presenting significant ecological and economic challenges. To mitigate these impacts, substantial volumes of *Sargassum* biomass are often removed from coastlines and deposited into nearby forest ecosystems. We investigated the ecological implications of *Sargassum* deposition using manipulative field experiments to simulate realistic biomass inputs in beach and forest ecosystems. Contrary to expectations based on ecological theory (i.e., the Home Field Advantage hypothesis), our findings reveal comparable decomposition rates between beach and forest ecosystems. Additionally, employing a "litter bag" experiment, we assessed the relative contributions of arthropods and microbes to *Sargassum* decomposition. Our results suggest that decomposition processes differ between ecosystems, with Telitrid amphipods driving decomposition on beaches and microbial communities dominating decomposition in forest environments. Furthermore, our study provides insights into the long-term effects of *Sargassum* deposition on nutrient cycling within these two ecosystems. We found that after 12 months, *Sargassum* may serve as a nutrient subsidy to native plants in the forest, albeit with slower utilization rates compared to non-native species such as Bermuda grass, *Cynodon dactylon*, found on beaches in the area. Overall, our findings challenge the notion that naïve ecosystems are incapable of processing novel subsidies, demonstrating the capacity of forest ecosystems to assimilate and utilize foreign organic matter.

*Keywords:*

**Introduction**

Population growth and urbanization deliver nutrients into coastal waters that stimulate algal blooms (Teichberg et al. 2010). The cosmopolitan, species-rich genera *Ulva* and *Sargassum* are responsible for the majority of macroalgae blooms (Ye et al. 2011, Smetacek and Zingone 2013, Xiao et al. 2021). Such blooms present economic and ecological challenges to impacted countries (Lapointe 1997, Teichberg et al. 2010, Smetacek and Zingone 2013, Rodríguez-Martínez et al. 2023). For example, macroalgal blooms can clog fishing nets, impede the passage of boats, and release unpleasant or harmful chemicals for humans (Teichberg et al. 2010, Smetacek and Zingone 2013, Resiere et al. 2021, Rodríguez-Martínez et al. 2023). Beach-cast macroalgal blooms can obliterate coastal life by creating anoxic conditions and shading benthic taxa (Hauxwell et al. 2001, Teichberg et al. 2010, Van Tussenbroek et al. 2017, Rodríguez-Martínez et al. 2019). Because seaweed blooms negatively impact coastal environments and economies, it is common for this biomass to be removed from shorelines and dumped into adjacent habitats. However, the impacts of massive quantities of decomposing biomass on dump sites remain unknown. There is a need for manipulative experiments to understand how such anthropogenic activities subsidize and influence communities in recipient habitats.

At this time, there is only a nascent understanding of the impacts of seaweed dumping on terrestrial habitats. For example, it has been long proposed that the use of untreated macroalgae that is autochthonous to coastal areas can be beneficial for soil stability and fertility (Pereira et al. 2019) and direct application of macroalgae as a nutrient source for agriculture is a traditional practice (Madejón et al. 2022). However, untreated seaweeds can have improper carbon-to-nitrogen ratios, complex polysaccharides, and additional salts that are not ideal for compositing (Dang et al. 2023), and can lead to increased salinity, fermentation, foul odor, the production of toxic metabolites for plants, leaching of nitrogen into groundwater, changes in pH, and the buildup of heavy metals when applied to soil (Madejón et al. 2022). Currently, residual biomass from hydrocolloid extractions (e.g., agar, alginate, and carrageenan) and removal of seaweed blooms contributes a significant amount of waste that is discarded in landfills (Dang et al. 2023). Indeed, seaweeds have properties that can improve plant growth when properly treated (Madejón et al. 2022) and can have strong fertilization effects on coastal ecosystems (Piovia-Scott et al. 2013), however, because of the lack of co-evolution with terrestrial organisms, the impacts of seaweed dumping in those habitats (e.g., forests) are unknown.

An important bloom-forming seaweed that likely impacts coastal economies and ecology is the pelagic seaweed *Sargassum* (consisting of a mixture of *S. fluitans* and *S. natans*, Gower et al. 2013, Rodríguez-Martínez et al. 2020, Chávez et al. 2020). Since 2011, the shores of Caribbean countries, West Africa, and the Gulf of Mexico are often smothered in *Sargassum* deposits that are regularly over 1m deep (Doyle et al. 2015) and can cover the entire length and breadth of beaches (as pictured in Chávez et al. 2020) (pers.obs.???). The extent of beached *Sargassum* can be impressive - the northern sector of the Mexican Caribbean is now estimated to receive volumes ranging from 10,000 - 40,000 m³ of *Sargassum* per kilometer of beach per year (Rodríguez-Martínez et al., 2023) with most landings happening during 5-7 months of the year (typically April-September).

Although floating *Sargassum* patches have been observed in the tropical and subtropical Atlantic Ocean since the 1800s ( Brooks et al. 2018, Uribe-Martínez et al. 2022), massive strandings of *Sargassum* that coat entire coastlines are a relatively new phenomenon. For example, the mean volume for *Sargassum* that landed during the peak month of 2019 (May = 6,403 m3 km-1) was greater than the peak months of 2018 (June = 3,699 m3 km-1), and 2015 (September = 2,360 m3 km-1) in the same sector (Rodríguez-Martínez et al. 2022). Several factors have been implicated in the increase in magnitude and frequency of *Sargassum* blooms including increased nutrient addition from deforestation and land-use changes (Wang et al. 2019), ocean warming (Wang et al. 2019, Johns et al. 2020), evolving upwelling patterns along the northeastern African coast (Wang et al. 2019), and mineral inputs from Saharan dust (Johns et al. 2020, Chávez et al. 2020). Furthermore, *Sargassum* bloom intensity and frequency is predicted to continue to increase (Smetacek and Zingone 2013, Rodríguez-Martínez et al. 2023).

*Sargassum* deposition on beaches has negative economic and ecological implications. *Sargassum* degrades pristine white sand beaches and turquoise waters that historically were major attractions for tourists to the Caribbean. Such tourism provided the basis for the majority of the Gross Domestic Product for many of these countries. For example, the Mexican state of Quintana Roo that attracted nearly 15 million tourists and generated an income of approximately US$10.8 billion in 2021 (Rodríguez-Martínez et al., 2023). As a result, increasing *Sargassum* deposition presents a significant obstacle for the entire region.

In addition to the economic impacts, there are negative ecological consequences of leaving *Sargassum* on beaches. Without removal of *Sargassum*, leachates, and organic matter resulting from decomposition induce a reduction in oxygen and pH levels, alongside increased turbidity, sulfur, and ammonia concentrations in coastal waters (Van Tussenbroek et al. 2017, Chávez et al. 2020, Rodríguez-Martínez et al. 2023). For instance, a significant *Sargassum* beaching event in 2018 was associated with a faunal mortality event, where hypoxic conditions led to the demise of 78 species of neritic fish, crustaceans, echinoderms, mollusks, and polychaetes (Rodríguez-Martínez et al. 2019).

To minimize the negative impacts along shorelines, humans deploy barriers offshore to catch *Sargassum* or utilize several strategies to remove *Sargassum* from beaches. These strategies involve a collection process that then requires the biomass be moved into adjacent habitats including beach dunes, forests, quarries, and garbage dumps (R. DeSantiago, R. Rodriguez-Martinez, J. Long, pers. obs.). Seasonal or periodic landings of macroalgae typically confer benefits to beaches by serving as a foundation for and contributing to the fertilization of coastal dunes, and serve as a resource subsidy to food webs (Polis et al. 1997, Huxel and McCann 1998, Anderson and Polis 1998, Marczak et al. 2007, Yang et al. 2010, Spiller et al. 2010, Williams and Feagin 2010, Wright et al. 2013, Piovia-Scott et al. 2013). However, Dumping massive amounts of *Sargassum* could impact recipient ecosystems via several pathways including 1) nutrient addition (Spiller et al. 2010, Piovia-Scott et al. 2013), 2) toxin release (e.g., *Sargassum* contains high levels of inorganic Arsenic, (Rodríguez-Martínez et al. 2020)), and 3) food provisioning for detritivores (Spiller et al. 2010, Piovia-Scott et al. 2011, Wright et al. 2013, Kenny et al. 2017). Unfortunately, the impacts of this large-scale “experiment” are largely uncertain for at least two reasons. First, the capacity of local communities that have historically received smaller amounts of *Sargassum* (e.g., beach dune communities) to keep pace, decompose and incorporate this material is unlikely given the sheer volume of material needing processing. Second, *Sargassum* is being dumped into some habitats that are completely naïve to seaweed subsidies (e.g., forests).

Ecological theory suggests that local adaptation through coevolution would result in faster decomposition of litter on its own soil over foreign soil, termed ‘*the home field advantage’* (Bocock et al. 1960, Gholz et al. 2000, Pugnaire et al. 2023). Thus beach-cast *Sargassum* should decompose more quickly over sand dunes than forest soil. Furthermore, terrestrial predator foraging in beach-cast macroalgae is a common feature in coastal ecosystems (Kirkman and Kendrick 1997, Rose and Polis 1998, Dugan et al. 2003, Colombini and Chelazzi 2003, Kenny et al. 2017), but less is known about terrestrial insect responses to this phenomenon. To our knowledge, this has not been tested with macroalgae deposits in forests. Moreover, previous work with *Sargassum* biomass showed a fertilization effect on plants on shorelines Bahamian islands and there have been efforts to use *Sargassum* as a fertilizer for vascular plants. Yet, the fertilization effect of *Sargassum* in forest has not been tested.

Here, we used manipulative experiments to examine the ecological consequences of dumping large amounts of *Sargassum* biomass in habitats currently used as *Sargassum* dumps (i.e., beach and forest). At both habitats, we created realistic piles of *Sargassum* with paired, unmanipulated controls, and surveyed plots quarterly for one year. We estimated pile decomposition by estimating pile volume and by measuring mass change of smaller amounts of *Sargassum* in mesh bags with different accessibility to microbes and arthropods. We surveyed soil respiration and soil ammonium, nitrate, and dissolved organic carbon levels. To study *Sargassum* impacts on the plant community, we surveyed vegetation on plots and adjacent to plots. To measure the impact of *Sargassum* on the arthropod community, we used pitfall traps and sticky traps to survey crawling and flying arthropods, respectively.

**Methods**

Between July 23 and August 2022 (hereafter, August 2022), we created *Sargassum* (~4.3 m3) piles in a beach and forest habitat in Puerto Morelos, Quintana Roo, Mexico. These habitats are commonly targeted as dumping sites and this region has been greatly impacted by increased deposition of *Sargassum* onto beaches since 2011 (Chávez et al. 2020, López-Sosa et al. 2020, Rodríguez-Martínez et al. 2023). All experiments were permitted by the Mexican government (CNANP-00-007) and the local property managers (Moon Palace Resort and Dr. Alfredo Barrera Marín Botanical Garden, for the beach and forest habitat, respectively). Both sites provide limited access to the public and therefore, should be minimally impacted by human disturbance. This region experiences a “hot subhumid climate with rainfall during the summer”, “a mean annual temperature of 27oC”, and “a mean annual precipitation of 1,105 mm” (Sánchez Sánchez and Islebe 1999) (Garcia 1973).

Because people dump *Sargassum* high on beaches and in beach dunes, our beach manipulation was deployed at the transition from the beach dune to the beach (20.99343° N, -86.82442° W). The beaches in this region are formed from calcareous sand derived from coral reefs and shells, as well as limestone from the Yucatan Peninsula (Castillo and Moreno-Casasola 1996, Mendoza-González et al. 2016) with altitudinal differences between the highest and lowest parts of the dune averaging 5 m on the coasts of Quintana Roo (Islebe et al. 2015). This coast is inhabited by ~237 plant species with varying gradients of salt-tolerant, low non-woody plants (<30 cm high) with cosmopolitan families (e.g., *Poacea*, *Astracea*, and *Leguiminosae*) as well as woody species at higher elevations (Islebe et al. 2015). Although dunes and vegetation on them are dynamic due to physical disturbance (e.g., hurricanes) and natural history of the area, common plants can be found distributed on these beaches. *Ambrosia hispida, Tournefortia gnaphalodes, Suriana maritima, Tribulus cistoides, Ipomoea pes-caprae, Sesuvium portulacastrum, Sporobolus virginicus, Canavalia rosea, Okenia hypogea, Croton punctatus Tribulus cistoides, Ernodea littoralis, Scaevola plumierii, Coccoloba uvifera Pithecellobium keyense*, *Cakile lanceolata, Erithalis fruticosa,* and *Ernodea litteralis,* are widely distributed in this area (Castillo and Moreno-Casasola 1996, Mendoza-González et al. 2016)*.* Further, we identified African Bermuda grass (*Cynodon nlemfuensis,* hereafter “grass”) throughout the site.

Because people dump *Sargassum* in old quarries in forest clearings (Rodríguez-Martínez et al. 2023) that allow access for large dump trucks, our forest manipulation was deployed on the perimeter of a forest clearing in a botanical garden (Jardín Botánico ECOSUR “Dr. Alfredo Barrera Marín”; 20.84400° N, 86.90278° W). This forest is 4.2 km away from known forest *Sargassum* dumps. The dry forests in this region are considered tropical semi-evergreen forests (Miranda and Hernández-X. 1963) that reach canopy heights of 15-20 m (Sánchez Sánchez and Islebe 1999). Characteristic trees in the forest include *Brosimum alicastrum*, *Talisia olivaeformis*, *Manilkara zapota*, *Myrcianthes fragrans*, and *Dideroxylon gaumeriq*. Characteristic understory species include *Drypetes lateriflora*, *Coccoloba diversifolio*, *Gymnanthes lucida*, *Thrinax radiata*, and *Coccothrinax readii* (Sánchez Sánchez and Islebe 1999). The forest soil type is Lithosol-Rendzina and thus has a well-developed topsoil, no subsoil, and are shallow (<30cm) over limestone (Islebe et al. 2015).

Our manipulation crossed Habitat (Beach, Forest) with *Sargassum* (addition, control). At each site, we haphazardly selected locations for paired plots (separated by 9 m between pile centers) and then randomly assigned one plot of each pair to either *Sargassum* addition or unmanipulated control (n=5). For *Sargassum a*ddition plots, we created large piles of *Sargassum* (~1.88 m radius) with a backhoe or wheelbarrows (for the beach and forest habitat, respectively). *Sargassum* piles were initially larger at the beach than the forest (5.25 ± 0.03 and 3.31 ± 0.09 m3, beach and forest, respectively; two-sample t test, t = 5.3058, df = 4.5363, p-value < 0.05). These pile sizes are within the range that humans currently dump in beaches and forests in this region (pers. obs.). All *Sargassum* used in this experiment was collected from drift *Sargassum* accumulated in the ocean at the offshore barriers installed by Moon Palace Environmental Services. As a result, terrestrial organisms (e.g., Talitrid amphipods) should have initially been absent from the experimental *Sargassum*. Beach plots were placed parallel to the water line.

To consider seasonal variation and successional patterns in the impacts of *Sargassum* dumping, we surveyed plots at deployment (August 2022 and three additional dates: November 2022, March 2023, and August 2023). During each sampling period, we assessed pile decomposition and the impact of these manipulations on edaphic conditions, soil respiration, plants, and arthropods. To sample the same locations within plots across sampling dates, we marked the plot center with a flag, and we placed a rope circle around this flag whose radius was the mean radius of *Sargassum* addition plots at the initial deployment date (radius = 1.88m). We confirmed the position of the center flag using multiple photographs of each plot before placing the rope circle.

To estimate decomposition of entire *Sargassum* piles, we measured pile volume at each time point. Pile volume was determined with the equation of an elliptic cone (*V=1/3 πabh)*, where a=length of the semi-major axis of the pile footprint, b=the length of the semi-minor axis perpendicular to a, and h=pile height. Because pile volume is confounded by water loss and we wanted to understand the relative contribution of microbes and macrofauna to decomposition, we also measured decomposition of smaller amounts of *Sargassum* (wet mass = 235 ± 0.4g) deployed in each habitat inside bags with small and large mesh openings (0.18 mm and 10 mm; n=10). Decomposition in small mesh bags should have been driven by microbes since these bags excluded larger arthropods, whereas decomposition in large mesh bags was the result of both microbes and arthropods smaller than 10mm.

Mesh bags deployed in the beach habitat were placed adjacent to the large pile manipulation. Mesh bags deployed in the forest habitat were placed in a nearby forested location (20.8636° N, -86.9138° W) from August-November 2022, because this mesh bag experiment was started prior to deployment of the large pile manipulation. However, all forest mesh bags were moved to the botanical garden adjacent to the large pile manipulation in November 2022. At each sampling date (i.e., November 2022 and March 2023), we measured *Sargassum* wet mass. At the final sampling in March 2023, we also dried *Sargassum* to directly measure final dry biomass. We estimated starting dry biomass by calculating a dry:wet mass ratio and multiplying starting wet biomass by this ratio. These measurements were then used to calculate decomposition as a percentage of the initial dry biomass.

In addition to the mesh bags providing a more accurate assessment of decomposition, the large mesh bags also provided the opportunity to quantify the invertebrates attracted to *Sargassum*. After initial deployment of mesh bags in August 2022, we collected them after 7 days, removed all arthropods, and weighed and replaced *Sargassum* biomass into the mesh bags before returning them to the sites. After 3 months we collected all bags and repeated the foregoing steps. Arthropods were all counted and identified to order. Unfortunately, due to vandalism, we were unable to recover the large mesh bags from the beach habitat in March 2023, at which point, we obtained final wet and dry mass and concluded this experiment.

To understand if *Sargassum* additions increased soil nitrate levels, we collected 50 mL sediment cores from all plots. To minimize disturbance to the *Sargassum* piles, sediment cores were collected ~30 cm towards the plot center from plot edges. We cleared overlaying *Sargassum* and leaf litter before collecting soil. Samples from August 2022, March and August 2023 were dried using the lowest setting of a conventional oven (3 hrs, 190⁰C) then kept at -80⁰C until transported to San Diego State University (SDSU) in August 2023 where they were kept at -20⁰C until analysis. Because we did not have access to a drying oven in November 2022, these samples were dried at SDSU in a 65⁰C oven for 6hr before the analysis in December 2023.

We conducted a Spectrophotometric Nitrate assay using a vanadium chloride method. To do this ~10g of sediment was added to 25mL of 0.5M potassium sulfate solution and incubated in a shaker for 1hr at 120rmp at 30⁰C then spun for 5min at 10,000rpm. Supernatant from samples (10mL) was added to 20uL of 5M sulfuric acid for preservation in 15mL conical tubes. Final concentration of sulfuric acid was 10mM for the assay. Samples were pipetted in duplicates into a 96 well plate using 60uL of sample, 60uL of 50 mM Vanadium III Chloride, 30uL 116 mM Sulfanilamide, and 30uL NEDD (N-1-naphthylethylenediamine dihydrochloride). The plate was incubated at room temperature for 30min and then absorbance was measured at A540 nm using a spectrophotometer. Nitrate derived Nitrogen was calculated using the formula [(dilution factor)(Nitrate X uM)(14.00 g/mol Nitrogen)(0.025L)]/ (soil dry weight grams).

Samples for March and August 23 have been delivered to Andy 2/9. Hoping to get data soon but proceeding without it until then.

To determine the impact of *Sargassum* additions on soil respiration, we used two separate techniques. First, we collected gas samples that were later analyzed for CO2 levels with Mass Spectroscopy. Gas samples were collected from inverted containers (1.87 L) placed ~30 cm towards the plot center from plot edges. Prior to pushing these containers into the sediment, we removed overlying detritus. Gas was collected from containers with vacutainers immediately after deployment and then one hour later. Second, in August 2023, we used a respirometer (model #) to obtain *in situ* readings of CO2 at the plot center and 30cm towards the plot interior from the initial plot circumference.

*Plant Community Impacts*

To assess the effect of *Sargassum* on plant cover directly on plots (i.e., plot interior), we surveyed percent cover using 0.5 x 0.5 m quadrats with a 100-point grid. In August and November 2022, we haphazardly tossed a marker over the shoulder onto the interior and placed a quadrat where the marker landed (n=3). In March and August 2023, we switched to random sampling (by randomly selected cardinal directions and distances from the plot center). For all plant surveys, we only recorded the top “canopy” layer of plants beneath the 100 points (supplementary Fig. 3, 4.

*Plot perimeter survey*

To assess the effect of *Sargassum* on plant cover beyond piles, we surveyed percent cover adjacent to the edge of all treatments in August 2022 and 2023. We placed a quadrat at the edge of *Sargassum* plots and an equivalent location on control plots (i.e., 1.88m from the plot center), at four opposing poles or points of tangency. We measured percent cover using the methods described in the section above. For two of these poles, we also surveyed plant cover at two additional distances extending away from the pile edge (D1 and D2 = August 2022, 0.75 and 1.5 m at both sites; November 2022, 0.52 and 1.27m at beach, 0.62 and 1.37m at forest; March 2023, 0.32 and 1.07m at beach, 0.43 and 1.18m at forest; supplementary Fig. 3-6). In March 2023, we did not see variation between sampling distances (ANOVA, F1=2.85, p=0.09) and decided to drop this portion of the survey in the interest of sampling efforts and time (supplementary Fig. 2).

Additionally, because Bermuda grass was so prevalent at the beach sites and there were no obvious patterns observed in the forest or with other plants in general, we collapsed all non-Bermuda grass plants into one category in the Forest (March 2023), and both sites in August 2023. Furthermore, the *Sargassum* treatments had decreased in volume and area footprint (November 2022), which meant that quadrats placed on the edge of *Sargassum* for the perimeter survey had been moving over sampling periods, thus in August 2023 we decided to sample at 1.88m from the centers of *Sargassum* piles in the same way we sampled control plots. Because the quadrats on the “perimeter” were actually sampling over the original footprint of *Sargassum* plots, we excluded them from analysis and figures, as they do not accurately represent edge effects.

*Arthropod Community Impacts*

To assess the effect of *Sargassum* on the crawling arthropod community, we placed yellow plastic cups (210 ml) at the edge of the lowest and highest points of each plot for 24 hours. Pitfall traps were buried flush with the substrate and filed approximately halfway with water and ~5 drops of dish soap to break the surface tension and prevent arthropods from escaping. Because we wanted to assess arthropod abundance at the same time for each habitat to reduce variability from deployment dates, we waited until all *Sargassum* piles were completed at the beach before installing pitfall and sticky traps. At the point of installation, two *Sargassum* piles had been complete for two days, two had been completed for one day, and the traps were deployed after the final pile was made. All *Sargassum* piles were completed in one day in the forest and arthropod traps were deployed on the same day. After 24 hours, we collected the entire contents of each trap, and then counted and identified arthropods to order.

To assess the effect of *Sargassum* on the flying arthropod community, we placed two double-sided sticky cards (127mm x 76mm, Catchmaster ®), attached to wire rods, on each plot. The sticky traps were placed ~130mm above the substrate or *Sargassum* piles, ~1m from the center to the “high” and “low” sides of the plot. After 24 hours, we photographed all sticky traps *in-situ* to process at a later time. Arthropods were counted on both sides of sticky traps and identified to order.

**Statistical Analyses**

To test decomposition of *Sargassum* piles, we used analyzed pile volume as a percent of starting volume using a Linear Mixed Effects Model (LMEM), with site and sampling trip as the main effects and their interaction. We included *Sargassum* pile number in the model as a random factor to account for differences in starting volume. To test if decomposition of *Sargassum* differed in mesh bags with and without arthropod access, we used a LMEM with treatment, habitat, sampling trip as the main effects and their interactions, and bag identification number as a random factor to account for repeated measures. To test if arthropod abundance in large mesh bags differed between sites over time, we used a Linear Model with arthropod order, habitat and, sampling trip as main effects with their interactions. To test the effects of *Sargassum* addition on edaphic conditions we conducted three linear models with ammonium, nitrate, or DOC as a response variable with treatment as the main effect with habitat and sampling trip as interacting factors.

To test the effect of *Sargassum* additions on plant cover on the interior and the perimeter of plots, directly compared percent cover of grass and “other plants, in *Sargassum a*ddition plots to paired controls by habitat at each sampling period. We used *Cohen’s d*, which quantifies the difference in means between two groups (i.e., control vs. addition), divided by the weighted average of the standard deviations between both groups to determine the effect size. Using this standardized of effect size allows for comparisons of magnitude of the effect between groups on a common scale (i.e., small d≈0.2, medium d≈0.5, and large d≈0.8 effect sizes).We analyzed the effect sizes of *Sargassum* addition on crawling and flying arthropod abundances using the same method.

All statistical analyses and visualizations were conducted using the R Programming Language (R Core Team, 2023). Linear models and LMEM were conducted using the R base and ‘nlme’ packages. Necessary assumptions were met for statistical analyses and post-hoc tests were conducted as needed.

**Results**

Dumped *Sargassum* decomposed in both habitats, as measured both via change in pile volume (Supplementary Fig. 1) and biomass loss in the mesh-bag experiment (Fig. 1). The majority of decomposition occurred during the first three months. After one year, pile volume decreased by >85%. Pile volume as a percent of starting volume differed between habitats in November 2022 (LMEM: Estimate = 2.020, Std. Error = 4.857, t-value 3.918, p <0.001), but this effect was not observed in later sampling periods (p=0.65).

In November, habitat interacted with mesh size to influence decomposition of *Sargassum* in mesh bags (LMEM: Estimate = 21.05, Std. Error = 5.973, t-value 3.525, p <0.001). At this time point, decomposition in the forest was similar for large and small mesh bags suggesting that microbes drove decomposition in the forest. In contrast, decomposition in the small mesh bags at the beach was only 51% of the decomposition in the large mesh bags, suggesting that detritivores between 0.18-10mm played a much more significant role at the beach. Consistent with the pile volume temporal pattern, most decomposition occurred in the first three months of the experiment (LMEM: Estimate -17.805, Std. Error = 6.352, t-value = -2.803, p < 0.005). Unfortunately, we lost the large mesh bags at the beach in March to vandalism.

In August, habitat interacted with site to influence total count of arthropods in large mesh bags (LM: Estimate = 43.77, Std. Error = 5.298, t-value = 8.262, p < 0.001). At the initial time point arthropod count was significantly lower at the forest compared to the beach treatment (Estimate = -28.50, Std. Error = 6.12, t-value = -4.66, p < 0.001), likely due to high abundances of Amphipods inside mesh bags. Amphipod counts were 62x higher than Arachnids and Hymenopterans combined at the beach and 2% higher than all arthropods in the forest treatment. In contrast, no Amphipods were found in the mesh bags during November sampling and higher abundance of arthropods at the beach was due to Arachnids (Estimate = 27.667, Std. Error = 8.652, t-value = 3.198, p < 0.001; Fig. 2).

*Soil nutrients*

Ammonium levels were consistently higher in forest soil compared to the beach habitat and peaked in both *Sargassum* addition and control treatments in August and November 2022 (Fig. 3A, B). However, there was no treatment effect on ammonium levels from *Sargassum* addition in either habitat (LM: Estimate =0.738, Std. Error = 3.889, t-value = 0.19, p=0.85). *Sargassum* addition increased nitrate levels in both habitats (LM: Estimate =55.887, Std. Error =12.61, t-value = 4.432, p < 0.001). Nitrate levels peaked in March 2023 in the beach and in November 2022 in the forest (Fig. 3C, D). Although dissolved organic carbon was overall lower in the beach compared to the forest (Fig. 3E, F), *Sargassum* addition significantly increased DOC at both locations (LM: Estimate =199.32, Std. Error = 56.93, t-value = 3.501, p < 0.001).

*Soil respiration*

Using gas collectors, we measured generally higher levels of CO2 production in *Sargassum* plots (Fig. 4A, B); However, our analysis revealed no statistical significance in CO2 production over an hour period between treatments (ANOVA: F=1.679, p=0.219), sites (F=0.953, p=0.348), or their interaction (F=0.025, p=0.878) in November 2022. We measured zero CO2 production in control plots and 0.156 ± 0.0557 grams of CO2 m-2 hour-1 in *Sargassum* plots at the beach (Fig. 4A). In forest treatments, control plots produced 0.122 ± 0.0536 g CO2 m-2 hour-1 and *Sargassum* plots produced 0.244 ± 0.148 g CO2 m-2 hour-1 (Fig. 4B).

There was no clear pattern in CO2 production using the respirometer in the center or edge of plots (Fig. 4C, D). CO2  production was not statistically significant between position of respirometer (F=1.770, p=0.192) and treatment was marginally significant (F=3.971, p=0.0549). Further, there was a significant interaction between position and site (F=13.532, p<0.001), thus, we analyzed sites separately. CO2 production did not differ between respirometer position at the beach (F=3.873, p=0.067) but was higher in *Sargassum* plots than controls (F=5.593, p=0.031. In *Sargassum* plots, CO2 production measured 1.43 ± 0.601 on the center and 0.266 ± 0.0509 on the edge, and 0.146 ± 0.0354 on the center and 0.124 ± 0.0175 of control plots )(Fig. 4C). CO2 production in the forest did not differ by treatment (F=0.089, p=0.769) or position (F=0.063, p=0.804). In *Sargassum* plots, CO2 measured 1.40 ± 0.310 on the center and 1.13 ± 0.335on the edge, and 0.986 ± 0.187 on the center and 1.38 ± 0.152 on the edge of control plots (Fig. 4D).

*Plant community impacts*

Initially, the beach contained low plant cover (5%, supplementary Fig. 3, 5). *Sargassum* additions increased grass cover to >75% (Fig. 5A, C). This effect began to appear in November but was most apparent in March and August 2023. In contrast, grass cover remained low in control plots throughout the year (<10%; Fig. 5B, D). Non-grass plants only displayed weak responses to *Sargassum* additions. Although the effect was weaker than directly in the plots, the effect of *Sargassum* additions at the beach spilled over to increase grass cover 3.8x just beyond the plot perimeters (after one year *d*=1.02, Fig. 6A, C). This spillover effect was taxon-specific as *Sargassum* additions at the beach did not strongly impact non-grass plants outside of the plots.

Similar to the beach, the forest floor contained low plant cover at deployment (4%; supplementary Fig. 4, 6). Unlike the beach, *Sargassum* additions in the forest did not increase plant cover for any taxon within plots (Fig. 5B, D). There was a slight increase in effect size in “other plants” in the beach habitat in March 2023 (Fig. 5D), this was likely the result of a plot that was covered by a *Cucumis* plant that was not rooted within the *Sargassum* pile. Although there was not a direct effect of *Sargassum* additions on forest plot, there was a spillover effect – *Sargassum* additions increased non-grass plants at plot perimeters (Fig. 6B, D).

*Arthropod community impacts*

Crawling arthropods in pitfall traps consisted of arachnids and hymenopterans, mainly spiders and ants at both sites, and Talitrid amphipods at the beach (Fig. 7A, B). Adding *Sargassum* at the beach immediately increased amphipod abundance 42x in August 2022 (Fig. 7A, B, E). After the initial sampling of plots in August 2022, amphipods were rare [only 1 amphipod was found in a control plot in November 2022 and 1 in a *Sargassum* plot in March 2023 (Fig. 7C)]. Additionally, *Sargassum* additions at the beach initially increased arachnids and hymenopterans (August 2022). This effect remained for hymenopterans during later sampling events but switched to a decreasing effect on arachnids (Fig. 7A, C, E). In the forest, *Sargassum* addition only weakly affected crawling arthropods (Fig. 7B, D, F). Importantly, no amphipods were found in forest pitfall traps during any of the sampling periods (Fig. 7 B, D).

Flying arthropods sampled on sticky traps at the beach consisted of dipterans and hymenopterans, mostly flies and wasps (Fig. 8A,C). On the initial deployment of the experiment, amphipods (21 ± 7) were also captured on sticky traps, presumably because they move via jumping. Beach *Sargassum* additions initially increased dipterans 23x more than controls– however, effects that tapered out at subsequent surveys. *Sargassum* additions also increased hymenopteran abundance – but this effect was not apparent until November and March. Similar to the beach, *Sargassum* additions to the forest increased both dipterans and hymenopterans. However, the dipteran effect in the forest was stronger and delayed.

**Discussion**

Dumping massive amounts of *Sargassum* biomass on the beach and forest had habitat-specific impacts. Interestingly, decomposition of *Sargassum* piles did not differ between habitats after the first quarter but was likely driven by different processes (e.g., soil microbes in the forest and arthropods in the beach). Most *Sargassum* decomposition occurred in the first quarter of both field manipulations (i.e., *Sargassum* piles and mesh bags), however, it is likely that much of the initial mass and volume reductions were due to water loss in both systems. Overall, we did not find strong evidence to support the *Home Field Advantage* hypothesis in this study. Though HFA predicts that leaf litter decomposes faster on familiar soil, we measured similar *Sargassum* decomposition rates between at the beach and in the forest.

Surprisingly, *Sargassum* decomposition in the forest was likely driven by microbes, despite the presumed naivete of the habitat. Our analysis revealed consistent trends in ammonium, nitrate, and dissolved organic content over time in forest soil, irrespective of treatment conditions (i.e., *Sargassum* addition versus control). However, notable fluctuations were observed, including a peak in nitrate content and a decline in ammonium levels in November 2022. Concurrently, we observed elevated CO2 production above the substrate in *Sargassum* piles compared to controls during this period. Moreover, high levels of DOC in the forest ecosystem may serve as a substrate for microbial activity, potentially fueling decomposition processes and nutrient cycling. Conversely, the lower levels of DOC observed in the beach habitat may reflect differences in microbial community composition or metabolic activity. (need to better understand DOC to talk about it here)

*Sargassum* did not increase in percent cover of non-grass plants on the piles in either habitat. However, we observed an increase in percent cover on the perimeter of *Sargassum* piles in the forest. This finding aligns with previous research by Spiller et al. (2010), who reported a fertilization effect on coastal shrubs (*Conocarpus* *erectus* and *Coccolaba* *uvicera*) following *Sargassum* additions on beaches. Similarly, we detected a fertilization effect on the perimeter of *Sargassum* piles after 12 months, albeit with a medium effect size (Cohen’s *d* = 0.56), but only in the forest habitat. The limited plant response on and near *Sargassum* piles suggests that either 1) *Sargassum* did not provide a limiting nutrient to the vegetation, 2) nutrient remineralization and retention in the soil may be low, or a combination of both. Interestingly, we detected a strong response by a fast-growing, early colonizing grass species in response to *Sargassum* addition.

*Sargassum* dumping significantly increased grass percent cover at the beach habitat not in the forest. Bermuda grass *Cynodon dactylon* is not native to the region but it is widely distributed globally (Kole 2011). Given that *C. dactylon* thrives in disturbed habitats, has a high tolerance for abiotic and biotic pressure, is quick to establish, and has fast biomass production (Kole 2011, Casler et al. 2015), *C. dactylon*’s rapid colonization of *Sargassum* piles was unsurprising. Grass cover increased by 72% on the interior of *Sargassum* plots by March 2023, compared to only 7% in control plots. Due to its rhizomatous dispersal, this effect extended to the perimeter of *Sargassum* piles by August 2023. Although HFA suggests that organisms that have co-evolved with leaf litter (i.e., *Sargassum* wrack) would have a stronger response to the subsidy, our study only revealed a strong plant response from a non-native grass. Because *C. dactylon’*s global distribution success is often attributed to its efficiency in removing excess nutrients from soil, increasing soil retention with its rhizomes, and its use in various sediment types (e.g., coastal sand), we posit that beach-cast *Sargassum* can facilitate grass growth if its nutrients are remineralizer and made available to plants.

*Sargassum* additions also attracted a significant increase of arthropods in the beach habitat. Initially, there was 42x more Telitrid amphipods in *Sargassum* addition plots pitfall traps compared to those in control plots. The amphipod response was also true for mesh bags with arthropod access on initial deployment (Fig. 2A). Moreover, there was a small but positive response by amphipods in March 2023, though abundances were much lower than in August 2022. We hypothesize that the diminished *Sargassum* effect in November may be attributed to the prevailing dry and hot climate during the sampling trip, which may have deterred Telitrid amphipods from actively foraging in the beach habitat. Additionally, while dipterans and arachnids exhibited a numerical response in August 2022, this effect waned over time. Conversely, we observed either low or no response from crawling arthropods in the forest habitat, although a significant increase in dipteran abundance was noted during the November 2022 sampling trip (BUT WHY). Telitrid amphipods appear to be the main contributors to decomposition of massive *Sargassum* dumps on beaches, even when deposited above the high tide line. This suggests that amphipods are capable of traversing considerable distances across sandy beaches, risking desiccation, to access and utilize *Sargassum* biomass.

Evidently, *Sargassum* decomposition is feasible in either habitat but is driven by different organisms. While we observed a significant numerical response by arthropods at the beach, the response was notably low in the forest. However, despite the limited arthropod activity, decomposition in the forest was still facilitated through the microbial pathway, which was surprisingly efficientl. It is important to note that our study did not assess the chemical contents of *Sargassum* leachates, including salts, pH, and heavy metals, which could have long-term effects on forest soils and groundwater. Moreover, a longer-term experiment coupled with plant tissue analysis would be beneficial to elucidate whether trees uptake nutrients derived from *Sargassum*. Additionally, considering the known concentrations of arsenic in *Sargassum* tissue (REF), there are reasonable concerns regarding its leaching into waterways. However, it remains unknown to us whether arsenic is absorbed into plant tissue or released into the environment within these systems.

The efforts to address these massive Sargassum landings, spearheaded by grassroots initiatives and local government, have been both significant and commendable. While researchers try to understand the ecological impacts of this phenomenon, it's crucial to recognize that the movement of Sargassum biomass into these ecosystems is largely driven by the necessity to sustain local economies. Currently, various entities and individuals are actively developing new technologies to intercept Sargassum mats and exploring potential uses for Sargassum tissue, such as fertilizer, animal feed, and raw building materials. It's worth noting that the Puerto Morelos municipality has taken measures by funding a designated Sargassum disposal site with a polyurethane liner, aiming to mitigate the runoff of leachates and minimize environmental impact. To our knowledge, this site was not being utilized in 2023. Indeed, our findings suggest that forest soil may have the capacity to break down Sargassum tissue, but the long-term consequences of potential nutrient and heavy metal contamination of groundwater when not contained, remain uncertain. Despite these efforts, the majority of Sargassum biomass continues to be discarded on beaches and in forests, underscoring the importance of understanding the ecological impacts of dump sites to develop sustainable solutions for this challenge. (is it too hand wavy or even adequate to suggest that mixing sargassum and soil over these plastic liners might expedite decomposition while retaining leachates?)

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**Figure legends**

Figure 1

Percent decomposition of initial dry mass of large (gray bars) and small (white bars) mesh bags at Beach (left column) and the Forest (right column) in November 2022 (A) and March 2023 (B). Error bars represent mean ± SE. Note that no values are reported for large mesh bags at the beach in March 2022 due to vandalism, the lack of a bar does not indicate zero decomposition.

Figure 2

Mean arthropod abundance in large (10 mm) mesh bags over sampling period (August and November 2022) in the Beach (left column, panels A, C) and the Forest (right column, panels B, D). Error bars represent mean ± SE.

Figure 3

Ammonium (A,B), Nitrate (C,D), and Dissolved Organic Carbon (E,F) content (mg) per kg of sediment over sampling periods (August and November 2022, March and August 2023) by site (Beach A,C,E and Forest B,DF). Light gray bars represent Control treatments and dark gray bars represent *Sargassum* treatments. Error bars represent mean ± SE. Note that no samples were collected at the Beach site in August 2022.

Figure 4

Grams of CO2 per m2 over an hour in Control and *Sargassum* plots at the Beach (A) and Forest (B) from samples collected with ‘gas collectors’ November 2022. Respirometer CO2 (units) readings from Beach (C) and Forest (D) Control and *Sargassum* plot centers (gray bars) and edges (white bars) in August 2023. Error bars represent mean ± SE.

Figure 5

Mean percent cover of plot interiors over sampling period (August and November 2022, March and August 2023) at the Beach (A) and Forest (B). Shapes represent grass (●) and other plants (▲) and colors represent Control plots (gray) and *Sargassum* plots (black).

Error bars represent mean ± SE. Lower panels show the effect sizes by sampling period associated with the panel above (i.e., panel C effect sizes associated with panel A, and panel D effect sizes are associated panel B). Effect sizes were calculated using Cohen’s d (*Sargassum* vs. Control) for grass and other plants.

Figure 6

Mean percent cover of plot perimeter over sampling period (August 2022 and August 2023) at the Beach (A) and Forest (B). Shapes represent grass (●) and other plants (▲) and colors represent Control plots (gray) and *Sargassum* plots (black). Error bars represent mean ± SE. Lower panels show the effect sizes by sampling period associated with the panel above (i.e., panel C effect sizes are associated with panel A, and panel D effect sizes are associated panel B). Effect sizes were calculated using Cohen’s d (*Sargassum* vs. Control) for grass (●) and other plants (▲).

Figure 7

Mean crawling arthropod abundance in pitfall traps over sampling period (August and November 2022, and March 2023) in the Beach (top row, panels A, B) and the Forest (middle row, panels C,D). Arthropod abundances are separated by Controls (left column, panels A, C) and *Sargassum* (right column, panels B, D). Effect size plots are associated with plots above (i.e., panel E effect sizes are associated with panels A and B, panel F effect sizes are associated with panels C and D). Effect sizes were calculated using Cohen’s d (*Sargassum* vs. Control). Shapes represent Amphipoda (●), Arachnida (▲), and Hymenoptera (■) and error bars represent mean ± SE.

Figure 8

Mean flying arthropod abundance in pitfall traps over sampling period (August and November 2022, and March 2023) in the Beach (top row, panels A, B) and the Forest (middle row, panels C,D). Arthropod abundances are separated by Controls (left column, panels A, C) and *Sargassum* (right column, panels B, D). Effect size plots are associated with plots above (i.e., panel E effect sizes are associated with panels A and B, panel F effect sizes are associated with panels C and D). Effect sizes were calculated using Cohen’s d (*Sargassum* vs. Control). Shapes represent Amphipoda (●), Arachnida (▲), and Hymenoptera (■) and error bars represent mean ± SE.

**Supplementary Figure legends**

**Supplementary Figure 1**

Sargasso pile volume loss over sampling periods (August and November 2022, March and August 2023) as a percent (%) of original volume calculated for sargasso treatments in August 2022. Individual dots represent replicates at the Beach (black) and the Forest (white).

**Supplementary Figure 2**

Percent cover of grass (top four panels) and other plants (bottom four panels) over sampling periods (August 2022, November 2022, and March 2023). Control treatments (light gray) and *Sargassum* addition treatments (dark gray) are shown at Distance 1 (closest to plot perimeter) and Distance 2 (furthest from plot perimeter. Error bars represent mean ± SE.

**Supplementary Figure 3**

Proportion of cover categories in plot interiors over sampling periods 1 (August 2022), 2 (November 2022), 3 (March 2023), and 4 (August 2023) in the Beach site. Columns designate treatment type (left is Control and right is *Sargassum*) and rows represent treatment block.

**Supplementary Figure 4**

Proportion of cover categories in plot interiors over sampling periods 1 (August 2022), 2 (November 2022), 3 (March 2023), and 4 (August 2023) in the Forest site. Columns designate treatment type (left is Control and right is *Sargassum*) and rows represent treatment block.

**Supplementary Figure 5**

Proportion of cover categories in plot perimeter over sampling periods 1 (August 2022), 2 (November 2022), 3 (March 2023), and 4 (August 2023) in the Beach site. Columns designate treatment type (left is Control and right is *Sargassum*) and rows represent treatment block.

**Supplementary Figure 6**

Proportion of cover categories in plot perimeter over sampling periods 1 (August 2022), 2 (November 2022), 3 (March 2023), and 4 (August 2023) in the Forest site. Columns designate treatment type (left is Control and right is *Sargassum*) and rows represent treatment block.

**Figures**

Figure 1. Mesh bag decomposition

A graph of different sizes of trees

Description automatically generated

Figure 2. Mesh bag arthropod counts

A white squares with black letters

Description automatically generated

Figure 3. Soil chemistry

**A graph of different types of sea salt

Description automatically generated with medium confidence**

Figure 4. Soil respiration

A graph of different types of trees

Description automatically generated with medium confidence

Figure 5. Plot interior percent cover and effect sizes

A graph of different types of data

Description automatically generated with medium confidence

Figure 6. Plot perimeter percent cover and effect sizes

A graph of different types of trees

Description automatically generated with medium confidence

Figure 7. Craling arthropod counts and effect sizeA graph of a graph of a number of different types of growth

Description automatically generated with medium confidence

Figure 8. Flying arthropod abundance and effect sizes

A graph of different types of growth

Description automatically generated with medium confidence

**Supplementary figures**

Figure 1. Pile volume

A graph of a number of months

Description automatically generated

Figure 2. Distance from plot

**A graph of different types of plants

Description automatically generated with medium confidence**

Figure 3. Plot interior percent cover categories (Beach)

A chart of different colored squares

Description automatically generated

Figure 4. Plot interior percent cover categories (Forest)

A chart with different colored squares

Description automatically generated

Figure 5. Plot perimeter percent cover categories (Beach)

A chart of different colored squares

Description automatically generated

Figure 6. Plot perimeter percent cover categories (Forest)

A chart with different colored squares

Description automatically generated